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Original Articles

Quantifying the climate change mitigation potential of China's furniture sector: Wood substitution benefits on emission reduction

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ABSTRACT

Wood and wood-based products have positive effects on climate change mitigation. These products can substitute for energy-intensive products made of conventional materials. The main objective of this study was to quantify substitution benefits of wood furniture for China via national-scale mitigation analyses. We selected a basket of representative furniture and estimated the emission reduction for two functionally equivalent products, which differentiate in wood intensity. The emissions avoided in each type of product were weighted by its share of the furniture market to ensure that the basket can represent the current situation. Then, avoided emissions were estimated for wood commodities contained in the products basket. Results demonstrated that wood materials exerted low environmental impacts. From the less to the more wood intensity products with the equivalent function, the proportion of energy-intensive materials decreased 24% while the proportion of their greenhouse gas (GHG) emissions decreased 34%. The overall displacement factor for the wood material was 2.67 kg CO_2eq/kg or 1.46 tC/tC. The findings highlight the importance of material selection in mitigating climate change in the furniture sector. It is urgent to use of the existing timber resources efficiently and pursue alternative schemes for emission reduction to achieve the goal of GHG mitigation.

1. Introduction

Climate change caused by increased greenhouse gas (GHG) concentration in the atmosphere is the most significant challenge to human beings in the 21st century. According to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change (IPCC), the global average temperature in 2003–2012 increased by 0.78 °C compared with that in 1850–1900 (IPCC, 2013). As the dominant factor causing climate change, the carbon dioxide (CO_2) concentration has increased by 40% since the pre-industrial times (IPCC, 2013). The continuous emissions of GHG is expected to intensify global warming, so developing low-carbon economies whose main features are low energy consumption, low emissions, and low pollution has become a consensus among various countries.

China is the world's biggest GHG emitter. In 2015, 10.7 billion tons of GHG, almost one third of the global emissions, are made by China (Olivier et al., 2016). To mitigate the climate change, China aimed at reducing the CO_2 per unit of GDP by 60% to 65% by 2030 based on 2005 levels (CINDC, 2015). China has great pressure on emission reduction. Thus, to achieve the emission reduction goals effectively, less energy-consuming and more environmentally-friendly products are needed in all high-energy-consuming sectors. The ambitious GHG reduction targets require to include all effective mitigation actions such as substituting harvested wood product (HWP) for GHG-intensive materials. Some previous studies focused on uses of HWP in construction, such as window frames (Abeysundra et al., 2007), doors (Knight et al., 2005; Abeysundra et al., 2007), floor coverings (Petersen and Solberg, 2003; Geng et al., 2017a). All these studies suggested reduced GHG emissions by using HWP to substitute for non-wood materials (Table 1).

As the largest HWP producer in the world, China's HWP have a great potential in GHG mitigation, though the mitigation potential is relatively small compared to the total GHG emission by this country, the world's largest GHG emitter (Geng et al., 2017a). Especially, as an important part of China's forest industry, wood furniture industry accounts for 40% of the world production, which is the largest in the world (FAO, 2016). Data from China Emission Accounts and Datasets (CEADs, 2017) show that the CO_2 emissions of energy consumption in the furniture industry in 2014 were 1,470,000 tons. The total GHG

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Table 1
Review of wood substitution effect, and different studies of the displacement factor (DF).

Country	Reference	Specific sectors	Boundary	Method	Displacement factor (tC/tC)
New Zealand	Buchanan and Levine (1999)*	Buildings	Cradle to gate	Single DF	1.05 to 2.1
Norway	Petersen and Solberg (2003)	Floor coverings	Cradle to grave	Single DF	5.05
USA	Knight et al. (2005)	Doors	Cradle to gate	Direct comparison	–
Sri Lanka	Abeysundra et al. (2007)	Doors, windows	Cradle to grave	Direct comparison	–
Sweden	Sathre and O'Connor (2010)	21 different international studies (buildings, furniture, flooring, doors, windows, etc.)	Review (different boundaries)	Meta analysis Single DF	0.8 to 4.6, with an average value of 2.1
German	Knauf et al. (2015)	16 product areas (buildings, furniture, flooring, windows, packaging, etc.)	Cradle to gate	Weighted DF	1.1 to 2.4, with a single factor of 1.50
Canada	Smyth et al. (2017)	6 end-use products (buildings, furniture, flooring, decking, etc.)	Cradle to gate	Weighted DF	0.45 to 0.54
China	Li and Altan (2011)	Buildings	Cradle to gate	Direct comparison	–
	Gong et al. (2012)	Buildings	Cradle to gate	Direct comparison	–
	Geng et al. (2017a)	Floorings	Cradle to grave	Single DF	0.17 to 3.11
	Geng et al. (2017b)	15 different international studies (buildings, doors, poles, etc.)	Review (different boundaries)	Single DF	0.25 to 5.6

* The displacement factors range from a low of 1.05 to a high of 15 (extreme scenario, thus excluded).

emissions of the furniture sector are the highest among all types of wood products, appropriately accounting for 36% of the total forest products emissions in China, occupying the more and more important position in China's manufacturing industry (Gu et al., 2014). Material development with low energy intensity and reduced emissions remains popular issues in the furniture industry. Wood and wood-based products encounter strong competition from their substitute materials, such as plastics, steel, glass, or aluminum, which have offered numerous options to customers. Wood materials, especially those from sustainable harvesting practices, present a significant opportunity for emission reduction (FAO, 2016). Several studies has supported for the carbon mitigation potential of sustainably managed forests. On one hand, wood is a renewable material; it has minimal impact on the environment compared with other GHG-intensive materials (Sathre and O'Connor, 2010; Chen et al., 2014; Smyth et al., 2017; Geng et al., 2017b). The substitution benefits of wood in reducing GHG emissions are cumulative and permanent (Sathre and O'Connor, 2010). On the other hand, wood's environmental advantages become increasingly prominent when the carbon stocks of HWP are considered (Pingoud et al., 2003; Ji et al., 2016; Geng et al., 2017a). According to the China National Furniture Association in 2012, 696.49 million pieces of furniture were produced, of which 35.57% were wood furniture. GHG mitigation opportunities exist by greatly increasing HWP use in the furniture sector in China (CNFA, 2012).

The furniture industry is an assembling industry, employing a mixture of raw materials, including wood, metals, plastics, textiles, leather, glass and many others (FAO, 2016). Given that the amount of material substituted by wood products varies with different end-use categories, it is difficult to generalize the potential substitution benefits of wood-based materials in furniture. González-García et al. (2011) quantified the GHG emissions of several representative wood-based products from the furniture sector and reported that metals, boards, and energy use contribute the most to the environmental impact among all the assessed products; their total contributions range from 40% to 90%. Cordella and Hidalgo (2016) screened and reviewed 82 documents to analyze key environmental areas and improvement options for furniture products. They reported that material production and supply are the two key stages that can exert the most influence on the life-cycle assessment (LCA) of furniture products; their contributions range from 88% to 98%. The environmental impact depends on the upstream processes so, in our study, we used a cradle-to-gate system boundary to compare HWP and non-wood materials GHG emissions in the furniture sector.

The emission reductions by HWP substitution refer to the reduced fossil fuel emissions generated from manufacturing HWP in an end use, compared to that from the production of alternative non-wood materials (FAO, 2016). Displacement factor, DF, defined as units of reduced CO_2 emissions per unit of wood biomass or carbon contained by HWP that substitute for non-wood materials, is often used to estimate wood substitution effects (Sathre and O'Connor, 2010; Geng et al., 2017b). If non-wood materials in a particular application generate a given amount of GHG emissions when wood materials for the same application generate a different amount, then DF is calculated as the difference in emissions divided by the amount of wood (Sathre and O'Connor, 2010). Most studies describe this method for the direct comparison of wood with non-wood products with the same functional units, a single DF (Buchanan and Levine 1999; Petersen and Solberg, 2003; Geng et al., 2017a). Sathre and O'Connor (2010) integrated data from 21 studies in a *meta*-analysis, the published DF values varied among the 21 studies, because of the differences in system boundaries, non-wood materials substituted, energy mix in construction materials and types of buildings. They produced an average DF ranged from 0.8 to 4.6 tC of emission reduction per tC in wood product (tC/tC), and an single average DF of 2.1 tC/tC for HWP used in place of non-wood construction materials. However, the average DF (2.1 tC/tC) with this estimation is not always applicable. To estimate the national-level mitigation potential of wood substitution, it is important to quantify the DF by major categories of HWP end uses in the country (Chen et al. 2018). Hence, Knauf et al. (2015) improved the method by considering wood distribution at a national level, which estimated a DF for Germany-made HWP. In their study, sixteen estimates of displacement factors that ranged from 1.1 to 2.4 tC/tC were volume weighted based on a material flow analysis, and a single DF comprising all product areas was then established to be 1.50 tC/tC. Most recently, Smyth et al. (2017) used the same method to estimate the emission differences as DFs for sawnwood and panels produced in Canada. They compiled a basket of end-use products, included buildings, residential flooring, furniture and decking, and the avoided emissions for the end-use products basket were weighted by the Canadian consumption statistics.

Assessment of the emission reduction effect of domestic wood furniture is conducive to the implementation of measures to promote sustainable options. In China, research on the mitigation benefits of HWP focused mostly on the carbon accounting approaches for estimating carbon stocks and emissions (Ji et al., 2016). Only few studies have been conducted on wood's substitution effects, or the effects of a specific forestry sector such as flooring manufacturing (Geng et al., 2017a; Geng et al., 2017b). Geng et al. (2017b) only considered flooring in wood and ceramic tile as research objects, not the mitigation potential of wood substitution at a national-level. Geng et al. (2017a) also reviewed the DF comparison of two specific products. In this study,

we try to apply the weighted DF method to determine DF for wood substitution in the furniture sector. Based on the application of this method in the furniture sector, we can also adopt it in other energy-consuming sectors to compare the mitigation potential among different sectors at a national level.

The study is conducted by the following steps: first, we estimated the size of China's furniture product market, selected key end-use categories in furniture market segments, and identified a basket of end-use products and their system boundary. Second, we estimated the reduction in emissions for two functionally equivalent products: one product is more wood-intensive than the other. Each furniture product was weighted based on their shares to ensure that the basket reflected current usage, and avoided emissions were estimated for the basket of end-use products. Finally, we determined the emission reduction effects in the furniture sector and DF for the wood substitution at national level. The present study of quantifying HWP's mitigation potential in China's furniture sector can promote policy that helps China to meet its emission reduction targets with more HWP substitution.

2. Materials and methods

2.1. Analytical framework

The study estimated the national level DF for wood furniture substitution in China using the LCA method. The LCA method is a suitable tool to assess the environmental impact of products and services, so it should be included in the decision-making process for sustainability (González-García et al., 2012; Karin et al., 2015). It involves a quantified assessment of GHG emissions in the entire life cycle from raw material extraction to material processing, manufacture, distribution, use, and disposal (cradle to grave cycle). When the goal is an LCA on the target product, the system boundary of the life cycle should be defined first according to the research object to allow for a detailed study of each unit process and an easy comparison (Begman and Bowe, 2011). Cradle to gate is an assessment as a partial product life cycle from raw resource extractions to the factory gates; in other words, the assessment considered the sum of the environmental effects of all upstream processes before the product is used (Bai et al., 2013; Sathre and O'Connor, 2010).

Sustainable forest management that aims to maintain forest carbon stocks, while providing a sustained yield of HWP and wood bioenergy, has the potential to significantly contribute to climate change mitigation. This paper focuses on the HWP substitution effects from sustainably managed forests to mitigate GHG emissions, thus we did not include the forest carbon dynamics. In connection with the production processes of various products, we generalized all production processes as several phases according to their similarity (Fig. 1) and defined the system boundary of the life cycle of the furniture materials. We used a cradle-to-gate system boundary to compare GHG emissions of HWP and non-wood materials, i.e., emissions associated with operations and/or maintenance of furniture and end-of-service disposal were excluded from the analyses. We assumed that the emissions from transporting the finished furniture to the consumer were the same for more wood-intensive one and less wood intensive one (Smyth et al., 2017). Furniture do not need any maintenance other than mending, and the data on regular cleaning are dependent on consumer behavior, the difference in maintenance between the two alternative constructions can be ignored. In the waste handling phase, different treatments and disposals of retired furniture materials have impacts on the mitigation potential of HWP substitution. There are no China-specific GHG analyses for the disposal stages for all materials by furniture type in detail. Estimating these emissions is difficult and can be greatly uncertain, thus, we excluded the phase from the analyses.

2.2. Furniture segment

2.2.1. Furniture classification for estimating material consumption

The objective of this study was to estimate the national level DF for wood substitution in China's furniture sector. The emission reduction for each end-use sub-category and their consumption shares at a national level are the key to the results. As a large HWP producer and trader, China's furniture manufacturing industry has two main directions: domestic and export markets. Although the share of the export market in the China's furniture industry is not small, the exported furniture will not affect the HWP shares of domestic consumption for different types of furniture. Thus, only the domestic market of furniture is estimated.

As a direct downstream of the real estate industry, the development of the furniture industry is closely related to housing types in the construction industry. Xu (2005) estimated that household furniture accounted for more than 50% of all furniture produced in China, and furniture used in non-residential buildings accounted for the remaining faction. The shares were approximately proportional to the shares of the gross floor areas of residential construction and non-residential construction of all constructed buildings in China. Same as in Xu (2005), we assumed the production fractions of different types of furniture were the same as the fractional shares of gross floor areas of different construction types (MOHURD and CCIA, 2017): household furniture accounted for 67.35%, office furniture accounted for 5.63%, commercial furniture accounted for 6.64%, and the rest was for other furniture. The shares of different furniture types were assumed to be the shares of HWP consumption for these types of furniture. Given that household furniture has the dominant share, we subdivided it into four segments for analysis: living room, dining room, bedroom, and kitchen

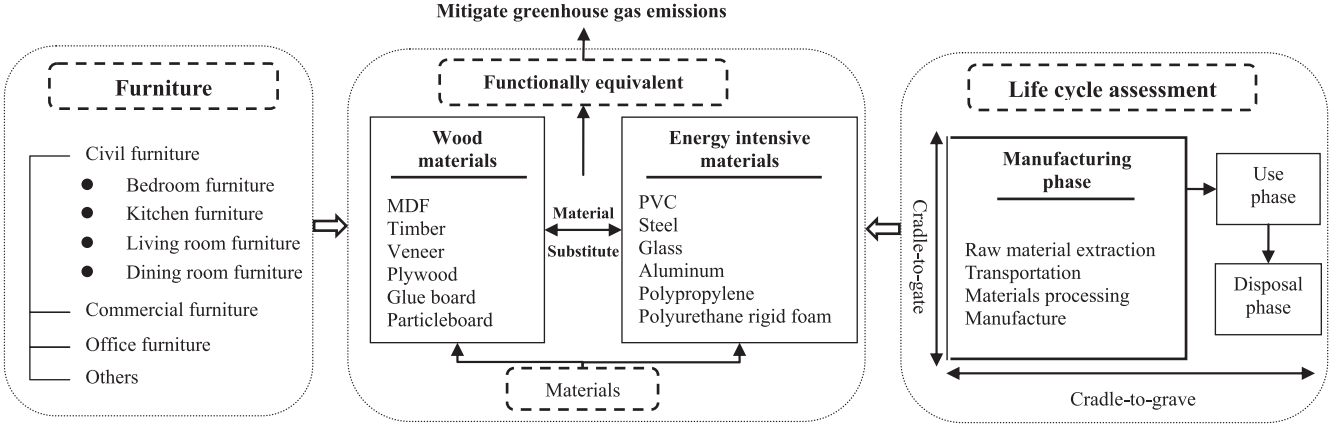

Fig. 1. Framework for evaluating the furniture sector wood substitution effect.

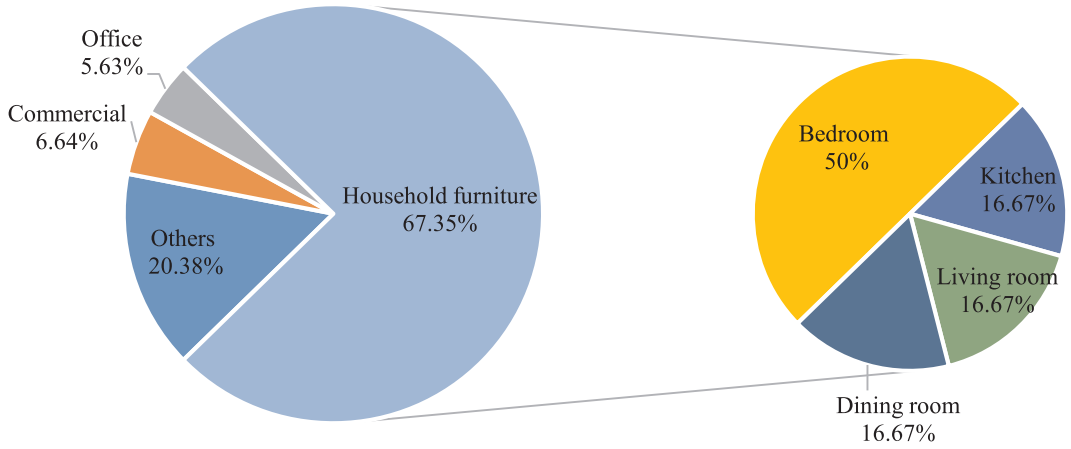

Fig. 2. Percentages of furniture production by type in China, which were used to estimate the domestic consumption of harvested wood products in furniture production in 2016.

furniture. The floor space of residents per capita in China was 40.8 m^2 in 2016 (NBSC, 2016). According to the China Family Development Report of the National Health and Family Planning Commission (NHFPC) in 2015, the average size of a Chinese family was 3.02 (NHFPC, 2015). Thus, the average floor space for each family is about 123.2 m^2, which is the area of a typical flat of three bedrooms, two halls, and one kitchen. Therefore, the following assumptions were made: (1) the proportions of Chinese household furniture stock is allocated according to the fact that each family has one set of living room furniture, dining furniture, and kitchen furniture, respectively, and three sets of bedroom furniture; (2) such proportions do not change significantly in the short term; and (3) consumers keep consuming this proportions of all the furniture, which implies new sales of furniture are roughly matched these proportions. With these assumptions, the shares of the household furniture segment market were determined as follows: bedroom furniture accounted for 50.00%, kitchen furniture, dining room furniture and living room furniture accounted for 16.67%, respectively (Fig. 2).

2.2.2. Basket of representative products

The furniture industry uses different raw materials to manufacture different products (Table 2). Given that the types and amounts of materials substituted vary by the application, national-level mitigation strategies require a broad scope and need the derivation of a DF for the primary wood material based on a series of end-use products (Smyth et al., 2017). The next step is to choose the representative products based on the key end use category as the functional unit. We selected the representative furniture that meets the function requirements of each end use category, such as bedroom furniture, living room furniture, etc. The kind of representative furniture was selected according to the China National Furniture Association which contains the China furniture industry market report. Then, we summarized the published consumption ranges of HWP and typical non-wood materials in furniture production by functional unit. The consumption of materials in furniture should be based on previous research reports, especially the domestic ones. When Chinese data are unavailable, we resort to other references whose research objects could meet the national furniture standard in the *China Furniture Almanac* (CNFA, 2016). The basket of furniture products included beds and wardrobes as the bedroom furniture, kitchen cabinets as the kitchen furniture; sofas, sofa tables, and TV chassis as the living room furniture; dining chairs as the dining room furniture; office chairs, office desks, and office cabinets as the office furniture; public space chairs as the commercial furniture; and student chairs as the others.

The amount of wood and other materials for furniture product differed between the more wood-intensive product and the less wood-intensive one. The main material of the kitchen cabinet was glue board; in its functionally equivalent products, we substituted timber and plywood for steel and plastic. The main material of the bed and the wardrobe was medium density fibreboard (MDF); we increased the proportion of MDF to replace hardware accessories, polystyrene foam board, and polyvinyl chloride (PVC) in their functionally equivalent products. Timber and MDF were the main integrating components of the sofa, sofa table, and TV chassis; we increased the proportion of MDF to replace glasses in the sofa table's equivalent product, steel and aluminum in the sofa's equivalent product, and plastic in the TV chassis' equivalent product. We increased the proportion of MDF to replace metal materials in the dining chair and reduced the proportion of metal materials in the public space chair. The main materials of the office chair, office desk, and office cabinet were timber, veneer, plywood, particleboard, and MDF; we increased their proportion to replace aluminum and plastic. Furthermore, we increased the proportion of particleboard, which has a low environmental effect, to replace metal materials in the student chair.

In the product basket which reflects national wood uses in the furniture sector, the main commodities in the wood furniture are sawn timber, wood-based panels, and other auxiliary materials (e.g., metal and plastic). The wood-based panels mainly consisted of fiberboard, particleboard, and plywood. Although the amount of non-wood materials, such as the hardware, do not account for a large share, these materials exert a great impact on the environment in their manufacturing phase and affect the environmental a lot. By compiling representative furniture and determining the reduction in emissions for two functionally equivalent products, we can determine the avoided emissions for each furniture product.

2.3. Methods

We used the weighted method applied by Knauf et al. (2015) and Smyth et al. (2017) to estimate the DF for wood substitution in China's furniture sector. This method was used to focus on DFs for wood substitution at the national level, and no one has applied it to a specific sector. For example, Smyth et al. (2017) selected several major HWP end use categories to reflect Canadian wood distribution, including buildings, residential flooring, furniture, and decking; then each end-use product was weighted by consumption statistics of Canada (Smyth et al., 2017). To estimate the wood substitution effects in a specific sector at national level more accurately, we applied the weighted method to the furniture sector in our study. First, we compiled a basket of representative furniture and estimated the reduction in emissions for two functionally equivalent furniture products. Second, each furniture product was weighted by the Chinese furniture market segments, thus

Table 2
Composition and associated material mass (in kilograms) of the compared furniture products for the functional unit basket.

Furniture			Materials	Less wood	More wood	Reference	Furniture		Materials	Less wood	More wood	Reference
Household furniture	Kitchen furniture	Kitchen cabinet	Glue board	28.40	36.52	Linkosalmi et al. (2016), González-García et al. (2011)	Commercial furniture	Public space chair	veneer	2.983	3.239	Linkosalmi et al. (2016)
			Plywood	0	14.76				plywood	0.34	0	
			Veneer	0	0.31				timber	0	0.023	
			MDF	2.10	0				steel	3.363	0.004	
			Timber	0.02	30.11				Polypropylene	0.015	0	
			Steel	11.80	0				Polyurethane rigid foam	0.423	0	
			Polypropylene	0.55	0		Office furniture	Office chair	Glue board	1.45	28.7	Spitzley et al. (2006), González-García et al. (2011)
	Bedroom furniture	Bed	MDF	54.35	162.50	Hoxha and Jusselme (2017), Jiang et al. (2014)			Particleboard	15.014	0	
			Steel	12.01	5.81				steel	22.861	33.0	
			PVC	0	3.287				Aluminium	12.7	0	
		Wardrobe	MDF	205.10	124.15	Iritani et al. (2015), Jiang et al. (2014)			Polypropylene	0.5	0.12	
			Steel	15.70	5.53			Office desk	veneer	0	5.0	Linkosalmi et al. (2016), Spitzley et al. (2006)
			Polypropylene	9.14	1.25				plywood	0	2.37	
	Living room furniture	Sofa table	MDF	23.17	41.60	Hoxha and Jusselme (2017), Jiang et al. (2014)			steel	14.650	14.25	
			Steel	5.27	1.58				Aluminium	6.078	0	
			Glass	22.24	0				Polypropylene	6.62	1.11	
			PVC	0	0.91				Polyurethane rigid foam	0	1.8	
		Sofa	Timber	10.88	22.90	EPD (2017a), EPD (2017b		Office cabinet	veneer	0	0.336	Linkosalmi et al. (2016), EPD (2017d)
			Steel	6.17	2.40				MDF	0	5.28	
			Aluminium	3.00	0				Particleboard	0	32.46	
			Polypropylene	8.09	0.30				steel	60.958	1.73	
			Polyurethane rigid foam	0	16.60				Aluminium	0.1	0.09	
		TV chassis	MDF		0.011	Smyth et al. (2017)			Polypropylene	0.062	2.34	
			Polypropylene	0.0034	0		Others	Student chair	MDF	1.11	0	Linkosalmi et al. (2016)
	Dining room furniture	Dining chair	MDF	0	2.60	Jørgensen (2013), EPD (2017c)			Plywood	4.5	0	
			Steel	2.20	0				Particleboard	0	6.38	
			Polypropylene	2.60	3.76				steel	6.86	6.069	
									Polypropylene	0.11	0.23	

obtaining the avoided emissions for each product within the basket. Finally, according to the percentage of wood used in each furniture product, DF was estimated for the wood material in the basket of end-use products. The specific calculation is as follows: we defined the less wood-intensive scenario and the more wood-intensive one, both of which were assumed to provide the same service. For each end-use product x, the reduced emissions E_x, for the more wood-intensive scenario relative to the less wood-intensive scenario was estimated as

$$E_x = \sum_{i=1}^{n} \Delta m_i \times \varphi_i \quad (1)$$

where n is the total number of materials in the end-use products x, Δm_i is the mass difference of i^{th} material between the two scenarios (kg), φ_i is the cradle-to-gate emission factor of the i^{th} material (kg CO_2eq of emissions per kg mass of the material, and $\Delta m_i \times \varphi_i$ is the emission difference (kg CO_2eq) associated with the i^{th} material between the two scenarios.

The weighted avoided emissions were estimated as E_{xD} (kg CO_2eq), which represents the avoided emissions for each product in the basket weighted by national consumption levels.

$$E_{xD} = f_x \times E_x \quad (2)$$

where f_x is the weighting factor for product x, it was estimated from the proportion of wood consumed at the national level for each broad wood use divided by the proportion of wood contained in the basket (Smyth et al., 2017). It was applied to weigh the avoided emissions of each end-use product in the basket. Moreover, f_x is calculated using Eq. (3):

$$f_x = \frac{P_x}{M/\sum_{X=1}^{12} M_x} \quad (3)$$

where P_x is the percentage of the national wood material consumption in the end use product x (%), M_x is the mass of wood material in the more wood-intensive end use product x (kg), 12 is the total number of the end-use products considered in the study, and $\sum_{x=1}^{12} M_x$ is the sum of the mass of wood material in the more wood-intensive end-use products (kg).

Then the weighted increments of wood used for the end-use product x were estimated as M_{xD} (kg):

$$M_{xD} = f_x \times \Delta M_x \quad (4)$$

where ΔM_x is the mass difference of wood material consumption between the two material assembly scenarios (kg).

Then, the weighted avoided emissions for the wood materials were divided by the incremental increase in wood mass for each product. DF was used to represent the substitution effect for wood as follows:

$$DF = \frac{\sum_{x=1}^{12} E_{xD}}{\sum_{x=1}^{12} M_{xD}} \quad (5)$$

It implies we divided the weighted emission reductions in the basket of end products by the weighted amount of the wood increment. The result is the amount of emission reductions achieved by each wood unit.

The final values of the DFs were converted from t CO_2eq/t wood to tC/tC, and calculations have been made by assuming the wood carbon is 50% of the oven-dry weight.

3. Results and discussion

3.1. GHG inventory

We focused on the main materials in the furniture products and used the GHG inventory data of these materials. Life cycle inventory (LCI) data are the primary inputs for conducting LCA studies (NREL, 2009).

In this study, the GHG inventory data of materials were obtained from existing studies of China as well as some international LAI databases.

We used GHG inventory data of steel, glass, PVC, aluminum, and timber in China, and their values are 2.6 kg CO_2eq/kg, 2.818 kg CO_2eq/kg, 4.65 kg CO_2eq/kg, 29.85 kg CO_2eq/kg, and 0.29 kg CO_2eq/kg, respectively (Wang et al., 2016). Obviously, materials as steel, plastics and aluminum invite higher GHG emissions compared to timber on an equivalent-weight basis.

However, the GHG or CO_2 inventory data of materials in China are limited, especially for the data of the wood panels. To estimate their inventory data, the average basic energy consumption and the conversion coefficients to standard coal are needed. Multiplying the energy consumption by the conversion coefficient of standard coal, we could obtain the CO_2 inventory data of these materials. According to the China's standard of forestry industry, the energy consumption is 220 kgce/m^3 for plywood, 140 kgce/m^3 for glue board, and 350.5 kgce/m^3 for MDF on average (Wu et al., 2016). The conversion coefficient of standard coal is 2.54 t CO_2/tce on average in China (Tu and Liu, 2014), thus, the estimated CO_2 inventory data of wood commodities contained in the products are: plywood is 0.745 kg CO_2/kg, glue board is 0.547 kg CO_2/kg, and the MDF is 1.225 kg CO_2/kg. The Ecoinvent database is one of the most well known LCI databases worldwide and is a database in terms of GHG emissions for the products in the cradle-to-gate phase. Except for MDF, the numbers of plywood and particleboard are similar to those in Ecoinvent database, so we adopted GHG inventory data of plywood, glue board, and particleboard from the Ecoinvent database, and their values are 0.766 kg CO_2eq/kg, 0.541 kg CO_2eq/kg, and 0.405 kg CO_2eq/kg, respectively. And we estimated the emissions of MDF based on the energy consumption in China. As polypropylene and polyurethane rigid foam account for small proportions of all materials, the GHG inventory data of the two materials were obtain from international databases directly.

As indicated in Table 3, the emission factors of metal, plastic, glass, and other energy-intensive materials were significantly higher than those of wood materials. Aluminum had the highest emission factor (29.85 kg CO_2eq/kg), followed by polystyrene foam boards (6.79 kg CO_2eq/kg). Large differences were also observed in the emission factors for different wood materials. The emission factor of veneer was the lowest. Particle boards showed lower environmental impacts than plywood and MDF, and the order of the emission factors of the three wood-based boards is consistent with the industry standard in China, in which MDF has the highest emission factor (1.23 kg CO_2eq/kg).

3.2. Substitution analysis

The selection of comparative end-use products may cause uncertainty (Smyth et al., 2017). In this study, wood material consumptions per functional unit in the less wood intensive scenarios range from 2.6 to 286.65 kg/unit (Table 2). In comparison, the increased wood consumptions per function unit of the HWP-intensive scenarios range from 0.061 to 57.68 kg. Multiplying the total mass of material consumption by the cradle-to-gate emission factors of these materials (Table 3), we could obtain the emissions (per function unit) in the more wood-intensive scenario and the baseline scenario.

For the two functionally equivalent products, wood materials amounted to 60% of sizes, metal materials accounted for 31%, and the rest were plastic, glass, and others in the less wood-intensive products. In the more wood-intensive products, wood material accounted for as much as 84% of their mass; the proportion of metal materials decreased to 11%. By determining the reduction in emissions for the two products on an equivalent-function basis, the results showed that metal materials were responsible for 68% of the global environmental impacts of the less wood-intensive furniture and 22% for the more wood-intensive one. The contribution of wood material increased from 24% to 58%. Based on seven furniture segments and their representative furniture (Table 5), the difference of cradle-to-gate GHG emissions of all materials between the two scenario range from 0.26 to 665.01 kg CO_2eq/unit, and the office furniture had the highest avoided emissions (655.01 kg CO_2eq). The avoided emissions for living room furniture (47.38 kg CO_2eq) were lower than those for the office furniture, and the lowest avoided emissions were for dining room furniture (small negative avoided emissions).

With the value of 655.01 kg CO_2eq, the office furniture had the highest avoided emissions and reduction potential. That is because the wood materials contained in the office furniture were substituted for aluminum and polystyrene foam boards that have high GHG emissions. The living room furniture also had steel and plastic being substituted, leading to 47.38 kg CO_2eq avoided emissions. The avoided emissions for dining room furniture were negative, the reason is that the plastic parts increased when wood was substituted for metal materials; thus, reduced emissions were negative. Though furniture avoided emissions were a small negative value, the overall displace factor regards the impact of all the end-use sub-categories.

We applied weighted avoided emissions for furniture to reflect China's national usage. The factors estimated in the functional unit basket ranged from 0.20 (bedroom furniture) to 22.45 (dining room furniture). The total HWP consumed for each end-use sub-category (M_x) could be obtained from Table 2, and the national total HWP consumption for furniture (P_x) could be obtained from SFA 2016 and Fig. 2. Based on the list of LCA studies on furniture provided in Section 3.1 and Table 4, the highest avoided emissions for end-use products weighted by national consumption were for the office furniture (258.24 kg CO_2eq) and living room furniture (128.46 kg CO_2eq), and the lowest avoided emissions were for the dining room furniture (−5.84 kg CO_2eq). Finally, the weighted avoided emissions for wood materials were divided by the incremental increases in wood mass for each product to obtain the DF for wood commodities contained in the products. By summing up the values of E_{xD} and M_{xD}, we obtained the total

Table 3
GHG emission factors for materials used in the functional unit basket.

Material	Emission factor (kg CO_2eq/kg)	Reference	Database
Steel	2.6	Wang et al. (2016)	China
Glass	2.818	Wang et al. (2016)	China
PVC	4.65	Wang et al. (2016)	China
Aluminum	29.85	Wang et al. (2016)	China
Timber	0.29	Wang et al. (2016)	China
Veneer	0.165	Bai et al. (2013)	U.S.LCI
Particleboard	0.405	Bai et al. (2013)	Ecoinvent
Glue board	0.541	Bribián et al. (2011)	Ecoinvent
Plywood	0.766	Bai et al. (2013)	Ecoinvent
MDF	1.225		China
Polypropylene	2.41	Bala et al. (2010)	
Polyurethane rigid foam	6.788	Bribián et al. (2011)	Ecoinvent

U.S.LCI: Life Cycle Inventory (LCI) Databases for the United States.

Table 4
Difference of mass and cradle-to-gate greenhouse gas emissions of all materials of each compared furniture products.

Furniture			Materials	φ_i (kg CO_2eq/kg)	Δm_i (kg)	E_{xi} (kg CO_2eq)	Furniture		Materials	φ_i (kg CO_2eq/kg)	Δm_i (kg)	E_{xi} (kg CO_2eq)
Household furniture	Kitchen furniture	Kitchen cabinet	Glue board	0.541	8.12	4.39	Commercial furniture	Public space chair	veneer	0.165	0.256	0.04
			Plywood	0.766	14.76	11.31			plywood	0.766	−0.34	−0.26
			Veneer	0.165	0.31	0.05			timber	0.2926	0.023	0.0067
			MDF	1.225	−2.1	−2.57			steel	2.6	−3.359	−8.73
			Timber	0.2926	30.09	8.8			Polypropylene	2.41	−0.015	−0.04
			Steel	2.6	−11.8	−30.68			Polyurethane rigid foam	6.788	−0.423	−2.87
			Polypropylene	2.41	−0.55	−1.33	Office furniture	Office chair	Glue board	0.541	27.25	14.74
	Bedroom furniture	Bed	MDF	1.225	108.15	132.48			Particleboard	0.405	−15.014	−6.08
			Steel	2.6	−6.2	−16.12			steel	2.6	10.139	26.36
			PVC	4.65	3.287	15.28			Aluminium	29.85	−12.7	−379.1
		Wardrobe	MDF	1.225	−80.95	−99.16			Polypropylene	2.41	−0.38	−0.92
			Steel	2.6	−10.17	−26.44		Office desk	veneer	0.165	5	0.83
			Polypropylene	2.41	−7.89	−19.01			plywood	0.766	2.37	1.82
	Living room furniture	Sofa table	MDF	1.225	18.43	22.58			steel	2.6	−0.4	−1.04
			Steel	2.6	−3.69	−9.59			Aluminium	29.85	−6.078	−181.43
			Glass	2.818	−22.24	−62.67			Polypropylene	2.41	−5.51	−13.28
			PVC	4.65	0.91	4.23			Polyurethane rigid foam	6.788	1.8	12.22
		Sofa	Timber	0.2926	12.02	3.52		Office cabinet	veneer	0.165	0.336	0.06
			Steel	2.6	−3.77	−9.8			MDF	1.225	5.28	6.47
			Aluminium	29.85	−3	−89.55			Particleboard	0.405	32.46	13.15
			Polypropylene	2.41	−7.79	−18.77			steel	2.6	−59.228	−153.99
			Polyurethane rigid foam	6.788	16.6	112.68			Aluminium	29.85	−0.01	−0.3
		TV chassis	MDF	1.225	0.011	0.0135			Polypropylene	2.41	2.278	5.49
			Polypropylene	2.41	−0.0034	−0.0082	Others	Student chair	MDF	1.225	−1.11	−1.36
	Dining room furniture	Dining chair	MDF	1.225	2.6	3.19			Plywood	0.766	−4.5	−3.45
			Steel	2.6	−2.2	−5.72			Particleboard	0.405	6.38	2.58
			Polypropylene	2.41	1.16	2.8			steel	2.6	−0.791	−2.06
									Polypropylene	2.41	0.12	0.29

φ_i: the cradle-to-gate emission intensity of the i^{th} material.
Δm_j: the i^{th} material mass difference between the more wood-intensive material assembly scenario and the less wood-intensive material assembly scenario.
E_{xi}: difference of cradle-to-gate greenhouse gas emissions of the i^{th} materials between the more wood-intensive material assembly scenario and the less wood-intensive material assembly scenario. $E_{xi} = \varphi_i \times \Delta m_j$.

Table 5
Weighting factors and displacement factors (DF) by end use category.

End-use categories	E_x	ΔM_x	M_x	M_x	P_x	f_x	E_{xD}	M_{xD}	DF_x	
Household furniture	(kg CO_2eq)	(kg)	(kg)	(%)	(%)		(kg CO_2eq)	(kg)	(tC/tC)	
	Kitchen furniture	−10.02	51.18	81.7	15.74	11.225	0.71	−7.15	36.5	1.45
	Bedroom furniture	−12.97	27.2	286.56	55.2	11.225	0.2	−2.64	5.53	
	Living room furniture	−47.38	30.461	64.51	12.42	33.675	2.71	−128.46	82.59	
	Dining room furniture	0.26	2.6	2.6	0.5	11.225	22.45	5.84	58.37	
Other furniture	Commercial furniture	−11.85	−0.061	3.262	0.63	6.64	10.57	−124.9	−0.64	
	Office furniture	−655.01	57.682	74.15	14.28	5.63	0.39	−258.24	22.74	
	Others furniture	−3.99	0.77	6.38	1.23	20.38	16.57	−66.11	12.76	

E_x: difference of cradle-to-gate greenhouse gas emissions of all materials between the more wood-intensive material assembly scenario and the less wood-intensive material assembly scenario by end use category. $E_x = \Sigma E_{xi}$ (based on the value of E_{xi} in Table 3).
ΔM_x: the mass (kg) difference of wood material consumption between the two material assembly scenarios (based on the value of Δm_i in Table 3).
M_x (kg): mass (kg) of wood material in the more wood-intensive material assembly scenario for the x^{th} end-use sub-category (based on the value of *More wood* in Table 1).
M_x (%): the proportion of wood consumed within the functional unit basket in more wood-intensive material assembly scenario (%).
P_x: the proportion of the national wood material consumption in the end use product x.
f_x: the displacement weighting factor for the x^{th} end-use sub-category. $f_x = P_x / M_x(\%)$.
E_{xD}: the avoided emissions for product in the product basket weighted by displacement weighting factor f_x. $E_{xD} = E_x \times f_x$.
M_{xD}: the weighted increments of wood used for the x^{th} end-use sub-category by displacement weighting factor f_x. $M_{xD} = \Delta M_x \times f_x$.
DF: displacement factors by HWP end-use sub-category (tC/tC). $DF = \Sigma E_{xD} / \Sigma M_{xD}$. Wood carbon content of is 50% of wood over-dry weight.

weighted avoided emissions for the representative furniture and the total weighted amount increment of the wood, and their values are 581.66 kg CO_2eq and 217.85 kg, respectively. The results showed that the overall DF for wood materials was 2.67 kg CO_2eq/kg or 1.46 tC/tC, meaning that for each tC in wood substituted in place of non-wood materials in the furniture sector, GHG emission reduction is approximately 1.46 tC on average.

3.3. Implications and limitations

In this study, the overall DF for the wood materials was 2.67 kg CO_2eq/kg or 1.46 tC/tC. This is comparable to the average DFs in the range of 1–3 tC/tC estimated by Sathre and O'Connor (2010), who also concluded an average DF of 2.1 tC/tC for HWP substitution. The results of this study are within Sathre and O'Connor's range, but lower than their average DF value. A reason is that Sathre and O'Connor (2010) integrate data from 21 studies in a *meta*-analysis, and the system boundaries are not same for all studies. Several studies involve forest ecosystem emissions and others involve postconsumer emissions from landfill and fossil fuel substitution. The system boundary in our study excluded forest growth, management, logging processes, and the end-of-life phase. Through reviewing the studies on wood substitution, it can be concluded that the DF increases as the end-of-life phase is included, especially when the fuel substitution effects of biofuels from wood processing residues or post-use wood products are considered (Gustavsson et al., 2006; Geng et al., 2017a). The other reason is that the mean value calculated by Sathre and O'Connor (2010 is not always applicable because their DF is only a difference of two specific products; large errors would occur if it is used to estimate DF at a national or regional level (Knauf et al., 2015).

Knauf et al. (2015) estimated the product DF for Germany. The range of their DF value is 1.46–1.62 tC/tC when solid wood is used as an alternative to glass, plastic, and metal in the furniture sector, close to the one estimated by our study. Smyth et al. (2017) estimated the product DF for Canada. The range of their DF value is 0.45–0.54 tC/tC when wood products are used as an alternative to other emission-intensive building products, lower than the DF estimated by this study. The range of DFs depends on the differences in non-wood materials substituted, system boundaries and national consumption statistics applied by the studies.

To accurately assess the effects of HWP substitution, it is important to include analysis of forest carbon dynamics of the no-harvesting and harvesting scenarios, HWP life-cycle carbon stocks and flows (from wood harvesting to the disposal of retired HWP). The adoption of a complete LCA is essential in promoting the accurate assessment of energy consumption and emissions from different products (Geng et al., 2017b). Especially, at the end of their service life, retired HWP undergo different disposal pathways: a) HWP can be recycled and reused to continue to substitute for non-wood materials; b) the retired HWP can also be burned with or without energy recovery and release wood carbon as CO_2; if the retired HWP are used to substitute fossil fuels, fossil fuel-based emissions can be reduced; c) retired HWP disposed in landfills decompose slowly, with about half emitted as methane (CH_4), but in a well-managed landfill facility, a fraction of the CH_4 can be captured, which can either be flared to convert to CO_2 to reduce the global warming effects, or burned to recover the energy to reduce GHG emissions (Chen et al., 2013; Geng et al., 2017a). Different treatments and disposals of retired furniture materials have impacts on the mitigation potential of HWP substitution. Future studies could focus on the proper handlings of wood waste.

Using wood harvested from sustainably managed forests to substitute for GHG-intensive materials can mitigate GHG emissions. However, with the rapid development of the China's economy, the demand for timber has become much greater than the growth of domestic forest resources. The logging ban and restrictive policies cut domestic timber supplies (Yang et al., 2010). The contradiction between the market demand and material supply has become an urgent problem. Therefore, the efficient use of existing forest resources is indispensable. The improved efficiency on utilizing small-diameter trees, inferior materials, and waste materials in the furniture industry is also required. It's necessary to pursue alternative emission reduction while achieving the sustainable use of forest resources.

The qualities of further studies could also be improved with better data collection and credible calculations with specific factors of material emissions. We partly obtained data from international databases. With more products being studied, the substitution effect of wood products would be better understood, and the emission reduction potential of the entire industry will be supported by more reliable evidence.

The environmental considerations gradually affect customer's choice of products, and the significance of green values is growing in the furniture sector (Linkosalmi et al., 2016). When purchasing a product, consumers are increasingly concerned about non-traditional issues, such as environment, safety, and sustainable products (González-García et al., 2011). Customers are demanding more specifically environmental information of the products (Linkosalmi et al., 2016). Knowledge regarding the environmental impact of materials used in the furniture sector is important to enable companies to improve their products from an environmental point of view (Bovea and Vidal, 2004). This study lays the foundation for further research.

4. Conclusions

In this study, we presented a general approach to quantify the mitigation benefits of wood substitution in furniture production. Based on the differences in consumption, we divided furniture sector into a few sub-categories and determined their weighted factor. The cradle-to-gate emissions of two functionally equivalent products were used to estimate the emission reduction. The weighted avoided emissions and the weighted incremental increase in wood mass of two functionally equivalent products were used to estimate the DF. The results highlighted the importance of material selection in mitigating climate change and reflected the unique characteristics of wood consumption in this sector of China. This study can serve as a reference to estimating DF for wood substitution at a national or regional level. Based on the above analysis, the overall DF for wood materials was 2.67 kg CO_2eq/kg or 1.46 tC/tC. In 2015, total consumption of wood materials in furniture in China was 66.99 million m^3 (SFA, 2016). Using the DF, if the total consumption increased by 1% on the basis of 2015 statistics, and the increased HWP were used to substitute for GHG-intensive materials, 24.45 Mt C of emissions can be reduced.

In addition to the furniture sector, construction materials also have great impacts on the mitigation potential of HWP substitution. Wood substitution effects have been intensively investigated in recent years, many new regulations and standards have been introduced in Europe to decrease GHG emissions (Linkosalmi et al., 2016). In China, *the 13th Five-Year Plan for the Development of Building* issued by the Ministry of Housing and Urban Construction requires the development plan of wood-structured buildings, and a special plan for the developing conditions of the wood structures. There is no doubt that policy guidance of wood material in China is clear and consistent with the direction of green development. At present, most materials consumed by the construction industry in China are energy-intensive; steel, concrete and cement account for more than 60% of the total energy consumption of all building components (Hong et al., 2016). Thus, through policy development that promotes HWP use in construction and non-construction applications, significant emission reduction can be achieved in China.

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